LCA Methodology

Extending the Life Cycle Methodology to Cover Impacts of Land Use Systems on the Water Balance[1]

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Abstract

Goal, Scope and Background. Whilst initially designed for industrial production systems, environmental life cycle assessment (LCA) has recently been increasingly applied to agriculture and forestry projects. Several authors suggested that the standard LCA methodology needs to be refined to cover the particularities of agri- and silvicultural production systems. Until now, water quantity received little attention in these methodological revisions, notwithstanding the well-known impact of agriculture and forestry on issues like water availability, drought and flood risk. This paper proposes an add-on to existing LCA methods in the form of an indicator set that integrates water quantity impacts of agri- and silvicultural production.

Method. First, system boundaries are discussed in order to identify the water flows between the production system and the environment. These flows are attributed to impact categories, linked to environmental burdens and to the areas of protection. Appropriate indicators are selected for each potential burden.

Results and Discussion. At the present, two input related impact categories deal with water quantity: Abiotic resource depletion and land use. The list of output related impact categories presented by Udo de Haes et al. (1999) does not include water quantity impacts like flood and drought risk. A new impact category 'regional water balance' is introduced to cover these risks. Exceedance probabilities are used as indicators for these temporal variations in streamflow.

Conclusion and Outlook. The method presented in this paper can bring a life cycle assessment closer to real world concerns. The main drawback, however, is the increasing data requirement that might hinder the feasibility of the method. Future research should focus on this problem, for instance by applying a relatively simple numerical model that can calculate the indicator scores from more easily accessible data.

Keywords: Agriculture; drought risk; flood risk; forestry; hydrology; land use; life cycle impact assessment (LCIA); water resources

[1] At present, a case-study dealing with catchment scale water quantity impact is not available. Therefore, this paper explores the potential benefits and difficulties of this subject in a theoretical way.

Introduction

Earlier research in life cycle impact assessment resulted in a diversity of impact categories and related indicators. Udo de Haes et al. (1999) synthesised the best available practice and proposed a structured list of impact categories to be used as a baseline (**Table 1**). In their overview, a discrepancy exists between the environmental importance of an issue and the detail it is dealt with in the impact assessment. One of the problems, as highlighted by Weidema (2000), is that the representation of some life cycle stages is out of proportion, while others do not get the attention they deserve. This trend is especially visible in sectors (food, wood, fibre) that entail agricultural or silvicultural production systems. The majority of LCAs is restricted to the industrial part of the production chain notwithstanding the potential environmental impact of biological production systems. This incited an ongoing revision of the existing methodology (Audsley et al. 1997, Mattson et al. 2000, Schweinle 2002) for incorporation of the impacts of agricultural or forestry practices.

The overall aim of this paper is to outline methodological improvements of available methods for water quantity issues, making a case for a spatio-temporally explicit approach. First, the system boundaries needed to define the water flows between the production system and the environment are discussed. These flows are then attributed to impact catego-

Table 1: Overview of impact categories as presented by Udo de Haes et al. (1999). Categories that contain water quantity issues are marked in italics

Input related impact categories
Extraction of abiotic resources
Extraction of biotic resources
Land use
Output related impact categories
Climate change
Stratospheric ozone depletion
Human toxicity
Eco-toxicity
Photo-oxidant formation
Acidification
Nutrification

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ries, linked to potential environmental burdens and to one of the four areas of protection (natural resources, ecosystem health, human health and man-made environment). Appropriate indicators are selected for each potential burden. The set of indicators can be used to make the impact modelling more realistic. This is of particular importance when a major part of the environmental impact is due to agriculture or forestry practices. Such activities have a significant influence on the regional water balance and, thus, on the risk of floods and droughts, two issues that are overlooked in the current practice.

1 Setting the System Boundaries

Hofstetter (1998) conceives the life cycle approach to environmental impact assessments as studying the interactions between three concentric spheres. The inner sphere or technosphere contains the product system. It is embedded in the ecosphere on which it exerts a certain environmental pressure due to emissions, waste disposal, etc. Overall, two kinds of interactions between the technosphere and the ecosphere can be distinguished: (I) extraction of inputs needed by the production system which are attributed to the input related impact group and (II) disposal of outputs produced by the system which are evaluated in the output related impact group. The outer sphere or valuesphere judges whether the environmental pressure caused by the production system holds an environmental threat. Bengtsson et al. (1998) propose a data model with a similar structure covering a technical, an environmental, a social and a geographical entity that mutually interact. The boundary between the eco- and technosphere needs special attention in case of location-specific assessments of agricultural and silvicultural production systems, because these systems are spatially and temporally dynamic (Ritter et al. 2001). For LCAs of industrial production, both spheres used to be considered as abstract static entities having no spatial and temporal scale (Karjalainen et al. 2001). In our case, the technosphere is a piece of land occupied for a certain time and surrounded (conceptually as well as physically) by the ecosphere, which is the subject of the impact assessment. So, for the period during which the biological production continues, the temporal fluctuations in site properties that are evaluated in the input related impact group are of no interest to the life cycle assessment as these belong to the internal management of the technosphere. When the biological production comes to an end, the technosphere dissolves yet it may still have long-lasting environmental effects. Input related on-site impacts are no longer trapped inside the technosphere so that the shift in site properties might be understood as an environmental impact. The magnitude of this impact depends on how the system boundaries are defined in time, i.e. when the production system is considered to end. The traditional approach evaluates the impact on water table height, for example, as the difference between the height at planting date and at harvesting date. So life cycles are assessed at the product level, for instance 1 kg of barley. The production of barley will often not stand on its own, yet will be part of a crop rotation system where a negative impact during one phase might be compensated later on. For example, lowering of the water table during a heavily irrigated phase can partly be compensated by lower water use during a fallow period. To account for such fluctuations of the environmental impact, it has been proposed to compare biological production systems in the perspective of one crop rotation (Cowell and Clift 2000). Input related impacts compare resource availability or site properties at the same phase within the cyclic production scheme, e.g. at the beginning of the rotation. Output related indicators address average impacts over one or several rotation periods (Karjalainen et al. 2001). Typically the impact of emissions will not be constant but will vary according to such land management practices as ploughing or clear cutting, and to external (climatic) conditions. The perception of the environmental pressure by the valuesphere sometimes depends on this temporal variability of emissions crossing the system boundaries. As a consequence, these impacts cannot be represented adequately by average indicators over a rotation period. This clearly is the case for water outputs: If all water is emitted at once, the flood risk will be higher than when water is released slowly. In addition to the expansion of system boundaries, this notion should be kept in mind when proposing an impact assessment method.

2 Input Related Impacts – The Impact Category Abiotic Resources

As depicted in Table 1, the abiotic resource and land use impact categories both handle input related impacts (Heijungs et al. 1997). The abiotic resource extraction category emphasises the reduced availability of the resource to future generations – in the case of water: The future freshwater reserves. Water issues may also turn up in the land use impact category that can contain quantitative aspects of land use (how much land is used for how long?) as well as qualitative aspects (Lindeijer 2000). Qualitative aspects constitute the so-called functional approach to land use impacts that addresses changes in the regulative functions of the land (Baitz et al. 2000). Land use systems may affect ecosystem functioning and, thus, water enters the land use impact category. As land use systems regulate the outflow of water, the ecosystem functions to be evaluated will be parallel to output related water impacts. Therefore, the integration of water in the land use impact category will be discussed later on in this paper after the section on output related impacts. Land use will still be considered as an input related impact though we first need to clarify a few concepts before elaborating water related land use impacts. The remainder of this paragraph will focus on abiotic resource depletion.

For the environmental impact of abiotic resource depletion, several approaches have been developed with the basic ones (summarised by Heijungs et al. 1997 and Lindfors et al. 1995) simply valuing the depletion according to the remaining reserves and the more sophisticated ones (summarised by Audsley et al. 1997) accounting for the pathway the resource follows after being released from the technosphere. One of the simplest indicator proposals of Lindfors et al. (1995), the static reserve life, will be used as a starting point for assessing the depletion of water resources. The indicator is defined as the ratio between the global reserve of the re-

source to the amount of the resource that is consumed. Hence, the indicator estimates the number of years that the activity can go on until the reserve is exhausted. Next to its simplicity, the major reason for selecting this indicator is that the depletion risk of all resources is expressed in the same unit, i.e. years, allowing one to rank all resources according to their risk to get exhausted. To become fully operational for water resources, the static reserve life concept needs two small modifications. The first problem is that resource reserves are estimated at a global scale. Cowell and Clift (2000) argue that such an approach, whilst useful for easily transportable resources, makes little sense for soil because soil losses in a certain region cannot be compensated by a reserve in another region. Since a similar objection can be made, at least partly, for water resources, estimating the available reserves at a smaller scale is proposed, i.e. a field or landscape unit. Doing so, spatial variability is included in the resource depletion assessment. Moreover, to become fully operational, a 'dynamic' reserve life is needed. The indicator of Lindfors et al. (1995) is static in the sense that it neglects new formation of the resource. For soil – or more general for fund resources – this makes sense because soil formation occurs at slow rates and the fertile upper layer of the soil that is prone to erosion is of a main concern. Since freshwater reserves are consumed and replenished much quicker then soil, assessing the sustainability of water use – or, more generally, assessing the consumption of flow resources – requires balancing water consumption with inflow. This results in **Eq. 1**:

$$Ind_A = \frac{R}{U - P} \qquad (1)$$

Where:

Ind_A: indicator of dynamic water reserve life (years)
R: freshwater reserves (mm)
U: water use (mm)
P: precipitation (mm)

When water use exceeds precipitation, the dynamic reserve life span will indicate the number of years until the freshwater reserves will be depleted, assuming that water inflow and outflow remain the same. This situation does occur for example in dry areas with heavily irrigated agricultural production like in the Middle East and Mediterranean countries (Yang and Zehnder 2002). The average annual rainfall in these regions varies between 0 and 340 mm per year, insufficient to meet crop water requirements. Another example are eucalypt plantations in southern India that transpire almost all water to be reached by their roots: These can consume more than 1000 mm of water per year, exceeding the average annual rainfall that amounts to 700 mm (Calder et al. 1997). If precipitation equals water use, the reserve life span becomes infinite, i.e. water use will never deplete the freshwater reserves. In case precipitation exceeds water use, the reserve life will be a negative value representing the number of years to get a precipitation surplus that equals the freshwater reserves available today. Note that a variable proportion of the water will reach the aquifer system while the remaining excess fraction leaves the catchment as runoff.

At first sight, temporal aspects could be a point of discussion in the assessment of the depletion of flow resources like water, because these resources can be temporally depleted depending on the timing of the resource use. The abiotic resource category is pointed towards the area of protection 'natural resources'. The environmental burden connected with this impact category is the availability of resources for future generations, not the competition for resources in the present generation, justifying the disregard of temporal variability.

3 Output Related Impacts

None of the output related categories in Table 1 deals with impacts on the water balance, although the potential burdens connected to water outputs are commonly acknowledged. To cover these burdens, a new impact category is introduced here, called 'regional water balance'. The areas of protection towards which this impact category is oriented are ecosystem health and human health, with flood risk, drought risk and average water availability downstream as main environmental burdens. A complicating factor, for the proper assessment of these impacts, is that one needs to consider the spatial organisation of the land use scheme as well as the timing of the emissions. The following sections explain how this can be accomplished and discuss the practical implications and the feasibility of the proposed method.

3.1 Spatial variability

Among LCA practitioners, there is a growing awareness of the value of site-specific data, at least for some output related impact categories (Ross and Evans 2002). For impacts on the regional water balance, spatial differentiation can be of great value since the control of water flows in an (agro-) ecosystem is location dependent. For example, the evapotranspiration of a forest will be different in southern Europe compared to Belgium, within Belgium in the loam belt versus the sandy region, within one region in the valley versus on the plateau. So it is evident that the impact of land use on water outflows cannot be judged without considering the spatial planning of the land use scheme. The best way to account for spatial variability in an LCA methodology depends on the decision-making context. Two cases can be distinguished: LCAs serve either as a guide for optimising a certain production process or as a means of selecting the most environmentally friendly option amongst different production scenarios. Similar classifications can be found in Hofstetter (1998), who distinguishes a static attribution case and a dynamic change oriented case, and Tillman (2000) who differentiates between a retrospective or accounting perspective and a prospective one. Both authors mention that the two approaches might lead to different scope definitions, inventory models and impact assessment methods. Tillman (2000) elaborated the implications of such a goal-oriented grouping for the inventory and so created a setting that was applied to our working field.

Identification of the critical subprocesses, i.e. the processes that contribute the most to the environmental damage, is the main task in accounting LCAs. Take for example the eco-labelling of agricultural products. Certain standards must

be reached in order to get an eco-label. A life cycle assessment can in this case help to demarcate the critical areas or parcels that have an extreme value for one or more indicators (high surface runoff rates, high soil erosion rates, etc.). Maps or GIS layers representing the spatial variability in indicator scores may satisfy the needs here as they indicate where the management should be adopted (by introducing zero tillage, etc…). The impact assessment is cause oriented, i.e. the main question is which spatial unit is causing which fraction of the impact. Indicator scores are seldom spatially aggregated and if so they are just area-weighed. Such an approach assumes that the total environmental impact is simply the sum of all the subunits – called 'additivity' by Tillman (2000), which is generally incorrect as complex interactions between subunits may raise or lower the total impact.

While useful for studying one particular scheme, the latter procedure is not suited to compare several scenarios. In this so-called static attribution case, the outcome of the impact assessment should be effect oriented, i.e. give an idea of the total effect per scenario. Assume that one wants to compare the environmental impact of traditional and organic farming to support the agricultural funding policy. The main question here is which scenario is the best, rather than which subprocess or spatial unit is causing the difference in impact. Indicators must therefore be represented in another way: One global score per scenario is needed instead of the GIS layers or maps. Applied to the impact category regional water balance, scenario optimisations ask for the contribution of each spatial unit to streamflow, whereas scenario comparisons are mainly interested in the total streamflow per scenario.

3.2 Temporal variability

It follows, no doubt, that making the temporal aspect explicit enhances the transparency of a methodology. This calls for multiple values for every variable of the indicator set, which can be gathered through long-term experiments or approximated with time series produced with a continuous model. Next, the time series must be aggregated into one or a few indicators reflecting the environmental burdens mentioned above. For average downstream water availability and drought risk, streamflow records averaged over a month seem appropriate because this is the highest level of aggregation where these impacts can still be detected. Then a quantile plot – showing the accumulated frequency of the observed or simulated monthly streamflow – can be constructed and a probability density function (PDF) can be fitted to the plot. Using the probability density function, the 5 and 50 percentiles can be calculated (**right part, Fig. 1**). The 50% quantile, or median, will replace the annual average score used earlier representing the average amount of water available downstream. This indicator has environmental as well as social importance, since it controls downstream ecosystem processes and the amount of water available for other human activities. The same remark applies to the fifth percentile, i.e. the monthly streamflow with an exceedance probability of 95%, which represents the risk of droughts. Peakflows cannot be derived from monthly aggregated flows, since a daily or even smaller time step is needed in this case. The 95th percentile or the daily streamflow with an exceedance probability of 5% can be used as an indicator for flood risk (**left part, Fig. 1**).

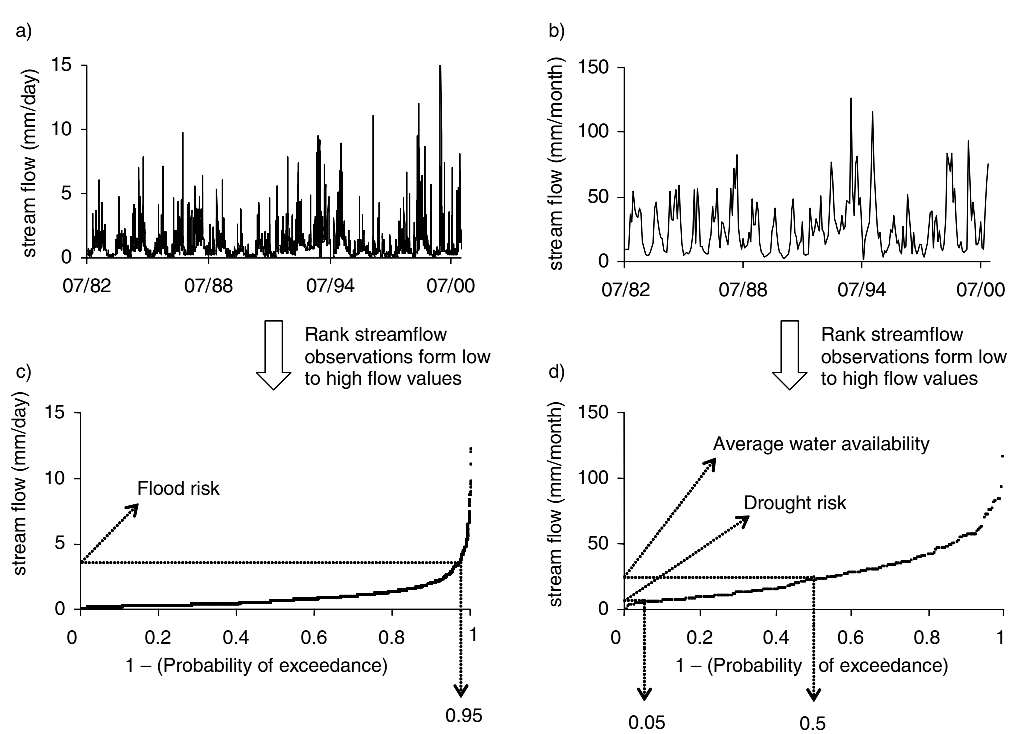

Fig. 1: Calculation of indicators for flood risk, drought risk and average water availability downstream. Theoretical example for the Maarkebeek catchment. Data are collected by the Flemish environmental administration (AMINAL)

3.3 Practical implications and feasibility

To calculate the proposed regional water balance indicators, streamflow records must be available. These data are not always accessible, so one could question the feasibility and universality of the proposed method. However, many hydrological models exist for estimating streamflow for a given land use scenario from more easily available data about climate, topography, soil properties and land use characteristics. Some models, e.g. the SWAT model (Soil and Water Assessment Tool) (Arnold et al. 1998), even include databases of crop characteristics needed to run the model. Because of this, data for a basic model application are available for almost every case study, though the accuracy of the modelling will depend on the quality and representativeness of the input data.

Only a limited post-processing of the output of a hydrological model is necessary to calculate the regional water balance indicators: It involves sorting streamflow observations from low to high flow values, eventually after rescaling the data to the appropriate time step, to calculate the median and the 5th and 95th quantile. The resulting indicators cannot directly be averaged into one single score because of a difference in magnitude – the drought risk indicator is always smaller than the indicator for average water availability downstream – and a difference in meaning – drought risk and average water availability indicators should be maximised, whereas flood risk indicators should be minimised. Moreover, the comparison of the indicators for different climatic zones may be difficult because of differences in the precipitation regime. The precipitation regime as well as other climatic factors constrain the flow regime and so affect the possible values of the regional water balance indicators. It is proposed to use the potential natural vegetation, which is site (climate and soil) dependent, as a reference system for making the indicator scores comparable. This gives us the following, normalised indicator formulas for average downstream water availability and drought risk (**Eq. 2**):

$$Ind_B = \frac{Ind_{B_{ref}} - Ind_{B_{act}}}{Ind_{B_{ref}}} \qquad (2)$$

Where:

Ind_B: normalised indicator of average downstream water availability and drought risk
$Ind_{B_{ref}}$: (non-normalised) indicator for the reference system
$Ind_{B_{act}}$: (non-normalised) indicator for the system under study

The normalised indicator for flood risk can be formulated as follows (**Eq. 3**):

$$Ind_C = \frac{Ind_{C_{act}} - Ind_{C_{ref}}}{Ind_{C_{ref}}} \qquad (3)$$

Where:

Ind_C: normalised indicator of flood risk
$Ind_{C_{ref}}$: (non-normalised) indicator for the reference system
$Ind_{C_{act}}$: (non-normalised) indicator for the system under study

Contrary to the non-normalised versions, this flood risk indicator has the same meaning as the drought risk and downstream water availability indicator, in the sense that positive scores indicate unwanted impacts and negative scores indicate desired effects. The normalised indicators can simply be averaged to get an overall score for the impact on the regional water balance.

4 Input Related Impacts Revisited – The Impact Category Land Use

Now that the method for assessing output related water impacts is established, let us revisit the qualitative part of the land use impact category. Several methodologies of varying complexity have been developed for assessing land qualities. The simplest ones are land use classifications that are too rough to assess the impact of land management. The more complex ones are functional approaches that reflect the impact of human activities on the functioning or the regulating capacity of the ecosystem (Lindeijer 2000). Some functional methodologies use a top down approach with one global indicator that summarises the total impact on ecosystem functions in a rather indirect way, e.g. the biodiversity indicator of Müller-Wenk (1998). Multiple indicators (e.g. Giegrich and Sturm 1998) may be preferable, however, for an intuitively more direct representation of environmental system complexity. It is with these 'multiple indicator' functional approaches that we meet water issues in the land use impact category.

Giegrich and Sturm (1998) used such a 'multiple indicators' approach reflecting the degree of naturalness of the land under the planned activities. Water resource indicators were chosen to represent water balance disturbances, e.g. artificial drainage, irrigation. Following their methodology, a negative impact is attributed to every human intervention so that an optimisation of land management is not feasible. Schweinle (2000) and Baitz et al. (2000) overcome this problem by evaluating the physical impact itself instead of the non-natural activities and structures that might cause it. Both authors describe the impact on water quantity with the variable groundwater supply, defined as precipitation minus surface runoff and evapotranspiration. Baitz et al. (2000) use one additional water quantity indicator called 'rainwater drain' that describes the ability of the land use to hold surface water. Schweinle (2000) mentions that all water balance terms are potentially useful indicators, apart from quantification difficulties.

As stated earlier, the 'function' of the land use with respect to water flows can be conceived as the way a land use system affects output related impacts, i.e. average water availability, flood and drought risks. The indicator groundwater recharge of the available method proposals can be coupled more or less to drought risk, and 'rainwater drain' to flood risk. Overall, land use affects the water balance through two mechanisms: By consuming a certain amount of water and by controlling how excess water runs off. Water consumption lowers water availability downstream and therefore

Table 2: Scheme of an LCA methodology for assessing impacts on water quantity

Impact category	Indicator	Environmental threat
Input related		
Abiotic resource depletion	Water dynamic reserve life	Future freshwater reserves
Land use	Change in surface runoff	Flood mitigating capacity
	Change in (infiltration minus evapotranspiration)	Drought mitigating capacity
	Change in precipitation surplus	Control on water flows
Output related		
Regional water balance	Daily streamflow with an exceedance probability of 5%	Flooding of human properties, disturbance of ecosystems by floods
	Monthly streamflow with an exceedance probability of 50%	Average water availability for other ecosystem processes and human activities, e.g. hydropower generation
	Monthly streamflow with an exceedance probability of 95%	Drought risk, drying of wetlands

influences local as well as regional ecosystem processes. It is proposed to quantify this environmental impact of land use with the indicator precipitation surplus that equals precipitation minus evapotranspiration. A part of the excess water infiltrates in the soil, possibly percolates to groundwater reserves, before it joins streamflow. The remainder forms surface runoff that reaches the channel quickly increasing the risk of high peak flows and floods. From an environmental impact point of view, distinction between these two flows is obviously necessary. This is achieved by adding a second land use indicator: Surface runoff. The amount of water that infiltrates the soil, diminished with the amount of water withdrawn by vegetation, can be used as an indicator associated with drought risk. For all three indicators, spatial variability should be handled in the same way as outlined before for output related impacts.

Whereas the water balance impact category evaluates the regional water balance during the activities – as a consequence of the outflow of water from the system, the environmental mechanism examined in the land use impact category is slightly different. This category analyses how the land use change and the land occupation that are part of the production system have altered the site properties so that the hydrological behaviour of the land changed. This change in land quality is evaluated as the change in water outputs of the land use system after one rotation compared to situation at the beginning of that rotation. As explained in the first paragraph on system boundaries, temporal fluctuations in land qualities during the production belong to the internal affairs of the technosphere. Consequently, they do not need to be evaluated in the land use impact category of a life cycle assessment.

5 Conclusions

Whereas earlier method improvements for LCA of land use systems focused on the extension of system boundaries leading to a better inventory, this paper deals with the spatio-temporal dynamics of flows passing the system boundaries, resulting in a more realistic impact assessment. **Table 2** shows an overview of the methodology constructed throughout this paper. Two input related impact categories are proposed, the abiotic resource category dealing with future freshwater reserves, and the land use impact category that is concerned with changes in the hydrological response of the land. For both categories, indicators are defined at a smaller scale compared to current method proposals in order to account for the spatial variability of water reserves and flows. The main step forward compared to existing method proposals is the introduction of a new impact category 'regional water balance'. This impact category covers a previously unexplored terrain: The output related water impacts, i.e. drought risk, flood risk and average downstream water availability. According to the goal of the study, indicators should be calculated per parcel if one wants to optimise a certain production scheme or per scenario in case of a comparison of several alternative production schemes.

Depending on the social context wherein the life cycle assessment is performed, the indicators from Table 2 might receive variable weighting. Although the methodology depicted in Table 2 can undoubtedly increase the credibility of the impact assessment, the main drawbacks are the increasing data requirements that might hinder the feasibility of the method. Future work should look for solutions to this problem, for instance by applying a numerical model to calculate the indicator scores from more easily accessible data.

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Excluding Site-Specific Data from the LCA Inventory:

How This Affects Life Cycle Impact Assessment

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The exclusion of site-specific data from the inventory phase of an LCA continues to be a point of controversy. Though the current simplified data collection strategy is widely supported by the LCA community, there are still many who are concerned about the implications this limitation has for the utility and reliability of LCA results. This is particularly relevant to practitioners who are attempting to draw conclusions about the environmental performance of different systems for the development of environmental policy. The current site-generic methodology introduces uncertainties into LCA results that have the potential to misdirect decisions on improvement measures. Therefore, in this paper we assess the practicality of collecting site-specific data and examine its value for study interpretation and decision-making. In our case study, we compare the contribution of a number of plastics-based packaging systems to photochemical oxidant formation. Our results demonstrate that the aggregation of photochemical oxidant precursor emissions into a single global parameter is an unreliable indicator of environmental burden and that the real significance of each packaging's contribution to the formation of photochemical smog in the atmosphere can only be understood after the addition of spatial and temporal information. We conclude that for non-global cumulative impact categories, additional spatial and temporal data should be collected, and that the benefits to decision makers far outweigh the additional effort needed to acquire this data for the LCA inventory.